SETAC-Europe: Second Working Group on LCIA (WIA-2)

Best Available Practice Regarding Impact Categories and Category Indicators in Life Cycle Impact Assessment

Background Document for the Second Working Group on Life Cycle Impact Assessment of SETAC-Europe (WIA-2)[1]

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Preface

The present report constitutes a basis for the identification of best available practice concerning impact categories and characterisation factors for Life Cycle Impact Assessment. It is the result of the first working phase of the second working group on Life Cycle Impact Assessment of SETAC-Europe. In this working group also members from other divisions of SETAC participated, in particular from the US and from Japan. The following members of the working group have contributed to draft versions of the document. Most, though not necessarily all, of these comments were taken over by the editorial committee:

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[1] For the publication in Int. J. LCA, the report has been divided into two parts. Part I includes Preface, chapters 1-3 and the complete list of references. Part II comprises Preface, chapters 4 & 5 and repeats the complete list of references. Part II appears in No. 3 (May 1999).

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1 Objectives

The priority aim of the SETAC-Europe working group on life cycle impact assessment (WIA-2) is to contribute to the establishment of best available practice regarding impact categories, together with category indicators, and lists of concomitant characterisation factors to be used in life cycle impact assessment. Such best available practice will enhance comparability between different LCA studies; at the same time this will stimulate a coherent scientific development. The establishment of best available practice regarding impact categories, category indicators and characterisation factors is to be distinguished clearly from standardised lists. Established best available practice is to be used as default; deviation is always possible and may well be highly recommendable in specific situations. Attention will be given to possibilities to work at different levels of sophistication, depending on different types of application. Also other generic differentiation in applications can be envisaged. In addition, the weighting between impact categories is included in the scope of the working group, be it with less priority in the work programme.

The first task of the SETAC-Europe working group is to start this process. This will also involve the identification of an international body that will provide an authoritative umbrella for this process, comparable to the work of IPCC on climate change under UNEP umbrella. If such a formal structure is achieved, the working group will continue by providing scientific input into this process. The working group aims at receiving relevant input from SETAC-North America and, if possible, from SETAC-Asia/Pacific during the whole process. The working period is three years, starting April 1998, but can be extended if necessary and possible.

The present document aims at presenting a consistent framework for the definition of impact categories and category indicators.

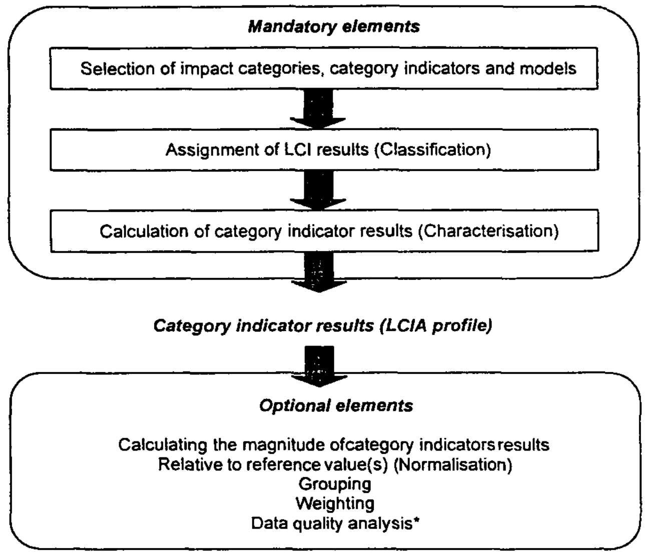
In **chapter 2** a closer description of the impact assessment framework is given, including the definition of a number of important terms and the establishment of a number of starting points for the work of the working group.

In **chapter 3** a number of general principles is discussed, underlying characterisation modelling.

In **chapter 4** an overview is given of impact categories, together with suggestions for possible category indicators (where in the environmental mechanism of an impact category?) and with an indication of their environmental relevance (relationship of category indicators with category endpoints).

The list of impact categories and the sections on general principles should, if possible, be decided upon as a basis for further work of the scientific task groups. The suggestions about category indicators are much more of a preliminary character. During the working period the proposals will further mature, as they will be tested against information and available methods analysed by the task groups dealing with the different categories.

The work of the new working group will build on the work of the first working group on Life Cycle Impact Assessment of SETAC-Europe (UDO DE HAES, ed., 1996), the work of the sister working group of SETAC in North America (BARNTHOUSE et al., 1997), and the analysis of their common basis and points of discussion (UDO DE HAES and OWENS, 1998).



*Mandatory in comparative assertions

Fig. 1: Elements of Life Cycle Impact Assessment (ISO/DIS 14042)

2 The Impact Assessment Framework

2.1 Technical framework for life cycle impact assessment

An important starting point for the work concerns the technical framework for life cycle impact assessment (LCIA) as defined by ISO/DIS 14042 (International Organization for Standardization, 1998), determining its required and optional elements (→ *Fig. 1*).

For our working group the most relevant element concerns life cycle impact assessment.

2.2 Definitions

A first term to be defined is *impact category*: a class representing environmental issues of concern into which LCI results may be assigned (ISO term, see *Fig. 2*). All physical processes and variables starting from extractions, emissions or other types of interaction between the product system and the environment, which are connected with a given impact category, are called the *environmental mechanism* of that impact category (ISO term). This replaces the term "cause-impact network" which was used by SETAC-Europe. In this terminology the environmental mechanism consists of a

number of environmental processes. Within and connected with this environmental mechanism one can distinguish:

- *environmental interventions*, such as in particular extractions from or emissions into the environment and other variables at the boundary of the product system and the environment, like different types of land use (SETAC-Europe term); note: the extractions and emissions are called together "elementary flows" or "environmental inputs and outputs" by ISO; we propose to use the encompassing term "environmental interventions"
- *category midpoints*, variables in the environmental mechanism of an impact category between the environmental interventions and the category endpoints (see below), like the concentration of toxic substances, the deposition of acidifying substances, the global temperature or the sea level (SETAC-North America term; in SETAC-Europe until now called "intermediate variables")
- *category endpoints*, being the variables which are of direct societal concern, such as human life span or incidence of illnesses, natural resources, valuable ecosystems or species, fossil fuels and mineral ores, monuments and landscapes, man-made materials, etc. (ISO term); the level of the endpoints is also called the "damage level" (SETAC-Europe term)

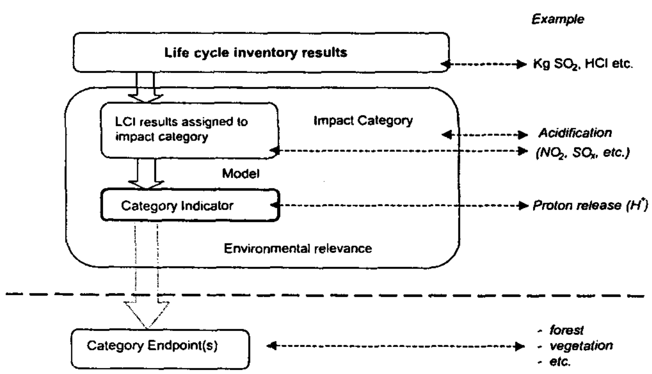


Fig. 2: Concept of indicators (ISO/DIS 14042)

- *areas of protection*, being classes of endpoints which have some well recognisable value for society (SETAC-Europe term; also called "safeguard subjects"); we distinguish four areas of protection: human health, natural resources, natural environment and man-made environment (the first three distinguished by ISO, the last added here).

Within the total of the environmental mechanism, a *category indicator* is identified, a quantifiable representation of an impact category (ISO term), being the object of characterisation modelling. Aggregation of environmental interventions within an impact category takes place in the units of the category indicator. The category indicator can be defined at any level of the environmental mechanism. The factors with which the environmental interventions are to be multiplied for this aggregation are called *characterisation factors* (ISO term) (sometimes called "equivalency factors"). The links between the environmental interventions and the category indicator are modelled as much as possible in a scientifically valid and quantitative way; the links between the category indicator and the category endpoints have to be identified either in quantitative or qualitative terms. These links are described with the term *environmental relevance* of the category indicator (ISO term).

2.3 Starting-points for the definition of impact categories and category indicators

In the ISO draft international standard a number of requirements are defined for characterisation modelling for "comparative assertions disclosed to the public", i.e. for public comparative applications of LCA. Furthermore, a number of requirements has been identified by the first SETAC-Europe working group (Udo de Haes, ed., 1996). Taking the input from these two sources together and adding also some new aspects, a list is given of relevant starting points for the definition of impact categories and category indicators and for the corresponding equivalency factor (→ *Table 1*).

Table 1: Starting points for definition of impact categories and category indicators

General starting point:	
1.	framework shall be developed which is open to further scientific progress and further detailing of information (new)
Starting points for the total of categories:	
2.	the categories shall together enable an encompassing assessment of relevant impacts, which are known today (completeness) (ISO/SETAC-Europe/new)
3.	the categories should have the least overlap as possible (independence) (SETAC-Europe)
4.	the total of the impact categories should amount to a not too high number (practicality) (SETAC-Europe)
Starting points for separate impact categories:	
5.	the category indicator can be chosen anywhere in the environmental mechanism of an impact category, from environmental interventions to category endpoints (ISO)
6.	the category indicator should (shall for comparative assertions) be modelled in a scientifically and technically valid way in relation to the environmental interventions, i.e., using a distinct identifiable environmental mechanism and/or reproducible empirical observation (ISO)
7.	the category indicator shall be environmentally relevant, i.e., it shall have sufficiently clear links to the category endpoints (ISO)
8.	it must be possible that characterisation factors are multiplied with mass or other units indicating the magnitude of the environmental interventions (new).

2.4 How to deal with value choices?

One of the most difficult points is how to deal with value choices in characterisation modelling. A core requirement in ISO/DIS 14042 concerns the scientific validity of category indicators, to be used in comparative assertions. This requirement is phrased in such a way that most of the present best practice of characterisation modelling may well be able to meet this criterion (UDO DE HAES and JOLLIET, subm.). It should be noted, that the category indicator can be chosen anywhere in the environmental mechanism, thus including the option of defining the indicator at endpoint level (as for instance years of human life lost, or YLL, based on combined modelling of fate and exposure and of effect). An important limitation, however, is that value choices are not allowed in comparative assertions. In order to better understand the meaning of this limitation, *value choices* have to be distinguished from *assumptions* (both ISO terms). Value choices are based on political, ethical, or ideological principles; assumptions are of a more technical nature and particularly will include choices that can in principle be empirically validated if sufficient information would be available. Value choices pertain for instance to the weighting between unrelated types of impact, such as impacts on natural environment vs. impact on human health, but also between impacts on different types of human health impairment. Assumptions for instance pertain to the data used on the background levels or on the persistence of chemicals in the environment. In ISO the choices in the modelling underlying GWPs and ODPs are all regarded as assumptions. In fact, this pertains to the choice of the time frame and the choice on whether or not to include data about background levels. In contrast to ISO, these should to our opinion be regarded as value choices, suggesting that in the ISO document value choices are considered acceptable if agreed upon by an authoritative international body.

The working group will take this ISO requirement as an important starting point. However, this starting point may in some cases appear to be at variance with other starting points, such as the choice for a surveyable number of impact categories for practicality reasons. In such a situation the working group may choose, if necessary, to deviate from the ISO requirement. This must be done in an explicit way, also addressing research needs in order to overcome this limitation in the future, if possible.

Another point here concerns the distinction between characterisation and weighting (ISO term; in SETAC: valuation). As indicated above, characterisation modelling should as much as possible be scientifically valid, although it may contain value choices as discussed above; on the other hand weighting across categories is inherently based on value choices, although it may also involve scientific input. A question is whether we may also accept real value choices in characterisation, if these are authorised by an international body. An example concerns the so-called disability adjusted life years (DALYs) developed under the auspices of WHO (MURRAY and LOPEZ, 1996). If so, the DALYs can, if desired, constitute the category indicator, but this option is at variance with ISO, although part of the DIS 14042 may suggest otherwise (see above). Therefore, it is preferable and more consistent to regard the disability adjustment as a generic weighting set, being part of weighting.

The limitation on value choices in externally used characterisation modelling does not imply that value choices are out of LCA altogether. The contrary is true, as is also emphasised by Barnthouse et al. (1997). In the Goal and Scope Definition phase value choices will determine the choice of the product group, the identification of the products to be included or excluded, the scope of the impacts to be investigated, etc. In Impact Assessment value choices can still be included for internal comparison or for external single claims. And in the Interpretation phase also value choices may play a role in the evaluation whether the aims of the study have been achieved or not. In this broader perspective, it is a question of how to manage value choices in a proper way, e.g., by using well defined procedures for review, rather than a question of an overall excommunication of value choices (cf. GOEDKOOP et al., 1998).

2.5 The choice of the category indicator

The choice of the category indicator defines the way in which equivalency between environmental interventions is to be established, i.e., the way in which they can be added up in common units. It may be useful to clarify this also in words through equivalency principles (JOLLIET, 1996) and not only in quantitative, formal expressions.

For each category it must be analysed what at present is the most proper place for defining the category indicator, taking into account the available scientific knowledge about the processes in the environmental mechanism of the impact category. Note: the same holds true for subcategories, if at stake, but this will only be mentioned if specifically relevant. As already mentioned, according to ISO all places in the environmental mechanism are allowed. In general, definition of an indicator closer to the environmental interventions will result in more certain modelling, but will render the indicator less environmentally relevant. In contrast, definition closer to the endpoints will make the indicator more environmentally relevant, but will render it less certain in its relationship to the environmental interventions (FINNVEDEN et al., 1992). Definition at the level of the endpoints themselves implies maximum environmental relevance. For an evaluation based on individual preferences a choice of the indicator at endpoint level seems even a prerequisite (NOTARNICOLA et al., 1998); it may also open new possibilities for science based aggregation (UDO DE HAES and JOLLIET, submitted). If an indicator is chosen at the level of the endpoints, the calculation of impacts on midpoints may still be of high interest and worth keeping as useful additional information in the analysis.

There is a tendency for defining the indicators further along the environmental mechanism, for the above given reasons. It may even be so that the logic long term aim will be to define all indicators at endpoint level (GOEDKOOP et al., 1998; HOFSTETTER, 1998). Because of the above mentioned possibility of science based aggregation, this may then also limit the number of categories to about three or four (cf. MUELLER-WENK, 1997). However, the working group aims to attune its recommendations to the current best practice, which may well result in indicators at different levels in the environmental mechanism. For instance, we can envisage the following possibilities:

- at the level of the interventions, e.g. kg of total material input or types of land use
- at midpoint level, e.g. climate forcing or proton release
- at endpoint level: e.g. years of life lost (YLL)

Choosing indicators at different levels necessitates very close attention for the consistency of the impact framework as a whole, avoiding as much as possible overlap between categories, or missing types of impact. It should be noted that this can be difficult if some indicators are defined at endpoint level and others at levels earlier in the mechanism. In categories (other than the human toxicity category) with indicators at midpoint level, which include in part impacts on human health, it should be tried in addition to calculate characterisation factors at endpoint level for these human health impacts. These can then be compared with the endpoint characterisation factors (if developed) of the human toxicity category. The same may well hold true for impacts on plant and animal species.

2.6 Categories and subcategories

The number of impact categories has to be limited by practicality. On the other hand there is a tendency to split categories because they are too heterogeneous and do not allow for scientifically valid aggregation. A possible way to cope with this problem is to define subcategories. Then the overall structure of categories remains surveyable, but the science-based aggregation takes place at the level of the subcategories, using subcategory indicators. This is beyond ISO but not in conflict with ISO. It implies that we will have value based weighting at two different levels: across subcategories within one category, and across categories. An example of weighting across subcategories within one category would be the already mentioned definition of DALYs: i.e. value based weighting between different human health subcategories within a broader human toxicity or human health impact category. But there may be many more examples (HOFSTETTER, 1998).

The structure to perform these two consecutive steps of weighting must be carefully considered from a decision theory point of view. It may be well advisable to aim at authorised generic weighting across subcategories in order to keep the weighting process in a given study surveyable. An example concerns the generic weighting procedure proposed for subcategories within human toxicity by Burke et al. (1995).

2.7 The areas of protection

As defined in Section 2.2, the areas of protection are classes of endpoints and are therefore of a physical nature. In that section it was also indicated that ISO distinguishes three areas of protection (although not under that term), i.e. human health, natural resources (providing options for extraction) and natural environment (with significance not related to extraction). In addition to that we distinguish here a fourth area of protection, i.e. the man-made environment. This new area of protection enables us to take damage to crops, to monuments or to materials like concrete into consideration. This inclusion is useful, because these types of endpoints can clearly be influenced by the environmental interventions caused by product systems. An example concerns the damage from acidifying substances to buildings. These four classes of endpoints are related to specific values in society like, for example, the intrinsic value of human life, the intrinsic value of nature, dealing with both ecosystems as well as species, cultural values and economic (or instrumental) values. In Table 2 an overview of the societal values related to the different areas of protection is presented.

Table 2: An overview of the relevant areas of protection together the connected main societal values

Areas of protection:	Societal values:
1. human health	Intrinsic value of human life, economic value
2. natural environment	Intrinsic value of nature (ecosystems, species), economic value of life support functions
3. natural resources	Economic and intrinsic values
4. man-made environment	Cultural, economic and intrinsic values

A special remark must be made about the inclusion of the man-made environment as an additional area of protection, because it may lead to an undesirable enlargement of the scope of LCA as an environmental tool. Careful description of the system boundaries must be considered and only those impacts of the product system should be considered, which involve environmental processes. Economic benefits or damages that are directly linked to the product function delivered are to be excluded. Examples are impacts of motor traffic on the economy caused by traffic jams, the increase of agricultural produce by the use of fertilisers or pesticides, or the loss or agricultural produce by changing agricultural land into nature area. These should be part of other tools like life cycle cost accounting and be performed separately from LCIA.

3 General Structure of Characterisation Modelling

Without going into any detail, we want to point out that there are some general principles for characterisation modelling which are very relevant for guiding the establishment of equivalency factors. These will be discussed in this chapter.

Table 3: Characteristics of a consistent fate and exposure analysis, for different types of effect coefficients

Level of Category indicator =>	deposition	media concentration	Dose	organ concentration	Organ or organism functioning
Relevant characteristics for equivalency =>	critical deposition rate	no effect concentration (1/NEC)	critical dose (or critical load) level	standard concentration in biota	critical damage to organ or organism
Fate (or fate and exposure) factor required for a consistent fate analysis =>	links emissions to an increase of deposition rate	links emissions to an integrated concentration increase	links emissions to an increase of intake or ingested doses	links emission to an increase in organ concentration	links emission to an increase in organ concentration

3.1 The modelling of fate and exposure and the modelling of effect

For the output-related categories it is imperative to include both modelling of fate and exposure, as well as effect. Fate modelling is regarded here as to include all environmental processes relating to the emission, transport, and transformation of the released substance; exposure modelling relates to all intake processes. A general condition is that both fate and exposure as well as effect should be included in characterisation modelling and treated in a consistent way. As described in the work of the first SETAC-Europe working group on impact assessment, this implies that the modelling of fate and exposure must link the environmental interventions to a specific category indicator (JOLLIET, 1996; see *Table 3*). For example, if human toxicity is characterised by a category indicator at the dose level, and using the critical dose for defining the characterisation factors, fate should relate the emissions of the product system to the dose taken in by humans through the different exposure pathways. But if human toxicity is characterised by organ concentration or by organ damage, fate and exposure together should relate the emissions to the concentration in the relevant organ, including the assimilation from mouth (or lungs) to that organ.

As indicated in ISO/DIS 14042, two approaches can contribute to quantitative characterisation: 1. a modelling approach (e.g. multimedia modelling) and 2. an empirical approach. This holds true for both fate and effect modelling. The empirical approach can play an important role as a check of the validity of the modelling approach (cf. JOLLIET and CRETTAZ, 1997). Up to now, fate modelling has been fully category specific. Yet, parts of the modelling will be relevant for more than one category only. Therefore consistency across categories has to be assured, possibly leading to across-category, or even generic modules for parts of the environmental mechanisms. For this reason also formal links must be achieved between the work which is being performed on fate modelling within the different scientific task groups.

3.2 Non-linearity of category indicators

The basic structure of LCIA implies that characterisation results are calculated as the linear product of inventory results and characterisation factors. This basic structure means that the results will be independent from the magnitude of the functional unit. The basis for this structure is that the impacts related to the functional unit are usually very small compared to the emissions leading to the background concentration (cf. POTTING and HAUSCHILD, 1997a). However, the magnitude of the characterisation factors may depend on the background concentration of a given substance, as they are linked to potential non-linearities in fate and exposure as well as in dose-response characteristics.

Some category indicators (being the dimension of the y-axis) are by their very nature linear because no further specification of the environmental processes, nor dose-response characteristics of exposed receptor organisms is taken into account. Examples are "proton release" or the burdening with macro-nutrients as category indicators. Furthermore, indicators where a given predicted concentration is divided by some reference concentration, (e.g. a risk quotient like PEC/NEC), do not take possible non-linearities of the underlying dose-response relationship into account. For both the characterisation factors are independent from the dose or concentration levels of the respective substances.

Other indicators are, with respect to their dose-response characteristics, by their very nature of a non-linear character. Examples are the percentage of individuals affected, as in usual bio-assays; or the number of species dependent on the concentration of a substance, e.g. the Potentially Affected Fraction of species or PAF-indicator (KLEPPER and VAN DE MEENT, 1997). The percentage of a given area which will be loaded above a given reference value also provides a non-linear curve, as a function of the background concentration and with different options to define such a reference value (POTTING et al. 1998a, b).

For the establishment of characterisation factors for non-linear indicators, a number of aspects play a role, for which further choices have to be made:

1. The first aspect concerns possible non-linearities in the fate (and possibly exposure) processes. These must be distinguished from differences between a high or a low stack height, or between indoor and outdoor, as these are examples of spatial differentiation and need not imply non-linearities. An example concerns the strong non-linearities

in tropospheric ozone creation; another example concerns the distribution of a macro-nutrients like ammonium, phosphate and possibly nitrate between soil and groundwater, depending on their background concentration.

2. The second aspect concerns possible non-linearities in the dose-response characteristics. If these are relevant, a working point has to be defined, particularly depending on the background level of the given substance. Each working point may then lead to its own characterisation factor. For deriving the characterisation factors from the dose-response function, a choice is to be made between a marginal and an average approach. Have we – for a given working point related to a given background concentration – to take the tangent of the curve (marginal) or the line connecting the working point with the 0-point (average), (see *Fig. 3*, where a sigmoid curve is taken as example)? A guiding procedure may well be to use the average approach for descriptive (i.e. not change oriented) questions, and the marginal approach for change-oriented questions; in general, LCA applications tend to be change oriented. Thus, the aim is in principle to develop two sets of characterisation factors for every category. In practice, however, it may be difficult to derive a marginal factor from the given data, which implies that then also an average factor will have to be used as an approximation. For both average and marginal approaches, the characterisation factors may be close or equal to zero at a range of low background concentrations. In addition, a marginal approach may imply that the characterisation factors are also close or equal to zero at very high background concentrations. Just as information: the GWPs and ODPs are derived in a marginal way from the underlying non-linear curves.

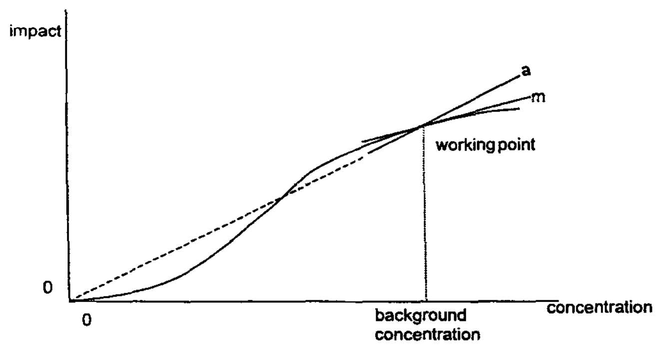


Fig. 3: Average (a) and marginal (m) characterisation modelling

3. A third aspect concerns the question whether also political thresholds have to be taken into account, in such a way that impacts at concentrations below that level are regarded to be zero, although in fact they are not. An example concerns the critical load level in acidification. In principle, such exclusion of politically acceptable situations should be avoided for reasons of scientific validity.

4. A last point is that the above process of deriving characterisation factors from non-linear fate and dose-response functions may also have to be done for different spatial as well as temporal conditions. These different condition may imply different working points, but may also imply site-dependent fate, exposure or other effect functions (see further section 3.4).

3.3 Temporal aspects

LCA essentially integrates over time. This implies that all impacts, irrespective of the moment that they occur, are equally included. There are also other options, however. Thus we may approximate infinite time by a given time period, say for instance of 500 years. The assumption then is that most of the impacts will have taken place and that the difference with infinite time can be neglected; an assumption which has to be verified. A quite different option is that we want to attach a lower weight to events in the far future, i.e. to perform some type of discounting. This can be done because future impacts are less certain and can perhaps be avoided; or just because we value the present higher than the past or the future. Discounting can be accomplished in two ways: either by a simple cut-off, as is done in climate change modelling by choosing for a 20, 50, 100 or 500 year period. Or it can be done by real continuous discounting of say 1 or 2% per year, (or 5 or 10% as is usual in economic analysis).

As best available practice it is proposed that, as a baseline, characterisation factors will be calculated for infinite time without discounting. If relevant, this can also be approached by a given long period of time, e.g. 500 years for global warming or 10.000 years for radioactive waste, as practical approximations. In addition, it must be checked whether a short period will yield considerably different results; if so, it is proposed that characterisation factors for a 100 year period will also be calculated. It has still to be checked whether 100 years is a useful short time period for all categories involved. Following this line, the start of the short time period (e.g. of 100 years) must be precisely defined, in particular for waste materials. Preferably the moment should be chosen that these materials reach the waste management stage and are out of reach of further direct human activity. Thus, best practice will then be expressed in a double result, thus enabling to attach different weight to events in the nearby and the far future (cf. FINNVEDEN, 1998).

3.4 Spatial aspects

Quite a number of applications may well need a spatially differentiated impact assessment. Examples concern regional differentiation for acidification (POTTING and HAUSCHILD, 1997 a,b), the distinction between urban and non-urban areas for photo-oxidant formation, or the distinction between indoor and outdoor emission of human toxic substances. The example regarding indoor and outdoor emissions indicates that not only a differentiation between regions is at stake. Spatial differentiation will increase the discriminating power

of LCIA; the need to proceed in this alley will increase if the systems to be compared are more alike.

Spatial differentiation may be performed because of differences in fate and exposure mechanisms, differences in sensitivity for effects, the difference between background levels determining the working point in the dose-effect curve, or, for resources, the difference between extraction below or above a sustainable level.

In general, the scientific task groups will start with a non-differentiated generic approach, aiming at global factors for every impact category. One step further concerns the development of spatially differentiated factors, where relevant, i.e. where large variations of fate and exposure or of effect variables are observed. In line with the aim of establishing generic lists of equivalency factors, the differences should be assessed by distinguishing between generic spatial classes. Spatial differentiation in the impact assessment phase requires attuning to the data gathering in the inventory phase, because the inventory data will have to be linked to the different distinguished regions. It is important that with spatial differentiation also the global characterisation factors will remain available, because the additional efforts on data acquisition in the inventory phase may not always be possible.

The way in which spatial differentiation will be performed will generally be category specific. However, for some categories the way of classification of region may have comparable requirements. The necessary attuning between impact categories can be a task of the scientific task group on fate modelling.

3.5 Uncertainty and levels of sophistication

Another question is how to deal with uncertainty about the processes in the environmental mechanism. In order to cope with this uncertainty, different levels of sophistication have first been distinguished in the Sandestin workshop on LCIA of SETAC (Fava et al., 1993). These levels were not yet consistently defined. A present aim is to define different levels of sophistication in various dimensions, for instance the fate and exposure, and the effect dimensions, and the temporal and spatial specification as modifiers within these dimensions (Udo de Haes, 1996; Jolliet, 1996; Heijungs and Wegener Sleeswijk, 1999; Potting and Hauschild, 1999).

One way to cope with large uncertainties, in particularly in the fate, exposure and the effect factors, can be to choose the indicator close to the environmental interventions (see above), accepting that the uncertainty will be moved to the environmental relevance of the category indicator. Thus, relatively certain indicators go together with a hidden uncertainty in their environmental relevance; this should be well documented. Another option may be to replace full quantitative modelling by empirical data for key parameters. Thus, we may explore the possibility to work with combined values of transport, persistency, bio-accumulation and toxicity factors, instead of working with a fully quantified and interdependent model, under the condition that the consistency of the information on fate and exposure and on effect is ensured. And there is the option of spatial and temporal differentiation, which will increase the accuracy of the results.

Because of the aim of LCA to describe the impacts of a product system in the most factual way:LCA focuses by its very nature on best estimates of the characterisation factors, not on worst case values. It is essential, however, that a quantitative estimation is made of the statistical variation of the characterisation factors by checking the model modules against experimental data, together with a qualitative discussion of possible other sources of uncertainty (including refraining from spatial and/or temporal information).

Another option for dealing with uncertainty in the fate and effect factors would be to work with an ordinal information, such as the data on persistency, bio-accumulation and toxicity gathered by US-EPA for 4000 chemicals (the so-called PBT approach, developed in the framework of the Waste Minimization Prioritization Tool). It should be noted that these ordinal data can not be multiplied with mass or other measures of the interventions, which is at variance with starting point no. 8 in section 2.3. However, this does not exclude the use of this information. Two options can in principle be taken up by the task groups:

1) the use of underlying quantitative information on fate and effect modelling (available at US-EPA under the heading of the draft prioritisation list at www.epa.gov/wastemin/)
2) the use of the PBT scores in a screening phase, as one may focus for instance on chemicals which have a 3-score for at least one of the three factors.

3.6 Application dependency

With reference to the above sections, the equivalency factors can be application dependent for at least the following three points:

1. The first point concerns the distinction between an average and a marginal approach.
2. The second point concerns the level of detail regarding possible spatial differentiation of characterisation modelling.
3. The third point concerns the level in the environmental mechanisms at which the category indicator is defined.

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